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A preliminary watershed scale soil quality assessment in north central Iowa, USA

Douglas L. Karlen[a,*], Mark D. Tomer[a], Jerry Neppel[b], Cynthia A. Cambardella[a]

[a] USDA-ARS, National Soil Tilth Laboratory, 2110 University Boulevard, Ames, IA 50011, USA
[b] Iowa Department of Natural Resources (DNR), 1305 E. Walnut, Des Moines, IA 50319, USA

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ABSTRACT

Soil quality assessment has been recognized as an important step toward understanding the long-term effects of conservation practices within agricultural watersheds. Our objective was to assess soil quality within the South Fork watershed of the Iowa River using various indicators and assessment approaches. Soil samples were collected during 2003 and 2004 from 29 areas of 32 ha (80 acres) each along two transects traversing the watershed. Soil pH, Mehlich III extractable P, K, Ca and Mg, electrical conductivity (EC), total organic carbon (TOC), and total N (TN) were measured. The Soil Management Assessment Framework (SMAF) was used to compute a soil quality index (SQI), while soil loss, the soil tillage intensity rating (STIR), N-leaching potential, and soil conditioning index (SCI) were determined for each sampling area using the 2003 version of the Revised Soil Loss Equation (RUSLE2). Overall, there were no soil fertility limitations within the watershed based on an average pH of 6.96 and extractable P and K levels of 36 and 162 mg kg^{-1}, respectively. Soil loss, STIR, N-leaching, and SCI averaged 1.13 Mg ha^{-1}, 68, 3, and 0.4, respectively. The SMAF analysis indicated soils within the watershed were functioning at 87% of their full potential. The lowest indicator score was associated with TOC (0.60) because the average value was only 28.4 g kg^{-1}. The SCI and SQI indices were positively correlated although since it used measured data, the SMAF appears to provide more information about the effects of management practices within the watershed. Soils in upper landscape positions had lower TOC and C:N ratios indicating an increased risks for both erosion and for nitrate leaching. Management of soils on hilltops may be the most effective way to minimize N and P losses within the watershed.

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1. Introduction

Assessments of soil quality at the watershed scale may be the link needed to demonstrate how agricultural management practices can influence water quality in streams (NRC, 1993). Potential soil quality issues include continued high rates of erosion, losses of organic matter, reductions in soil fertility and productivity, as well as chemical and heavy metal contamination (Larson and Pierce, 1991; Doran and Parkin, 1994; Karlen et al., 2001, 2003). The need to assess conservation effects led Andrews et al. (2004) to develop a Soil Management Assessment Framework (SMAF). They demonstrated that the tool could be used effectively for a variety of climates, soil types and soil management practices.

This study was initiated to evaluate nutrient management for the South Fork of the Iowa River watershed, but subsequently provided an opportunity to compare various soil quality assessment tools. The watershed is associated with USDA's Conservation Effects Assessment Project (CEAP) and located in north central Iowa, USA. The overall CEAP project was designed to quantify soil and water

* Corresponding author. Tel.: +1 515 294 3336; fax: +1 515 294 8125.
E-mail address: Doug.Karlen@ars.usda.gov (D.L. Karlen).

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quality at the watershed scale at several locations throughout the USA, but with an initial focus on water quality monitoring.

1.1. South Fork watershed characteristics

The South Fork watershed drains approximately 78,000 ha and is a relatively young landscape developed from recent (10^4 YBP) glacial deposits. Natural stream incision and development of alluvial valleys have occurred only in the lower parts of the watershed. Highly erodible land (HEL) occupies about 13% of the watershed (National Cooperative Soil Survey, 1985, 1986). Soil erosion can thus be a source of sediment in the streams. The upper parts of the watershed are occupied by till plains and marginal moraines that have many internally drained "prairie potholes." Hydric soils occupy about 54% of the area making soil wetness a major concern for land management and agricultural production. To solve this problem, artificial subsurface (tile) drainage was first installed more than 100 years ago, and today nearly all prairie potholes have been drained to a network of ditches that convey water to natural stream channels. Drainage has significantly increased agricultural production, but the subsurface tile and dug ditches have impacted water quality by decreasing surface water storage and hastening the routing of water from the watershed. Monitoring results (Tomer et al., in press-a) show significant quantities of NO_3-N, total P, and sediment in streams throughout the watershed. These pollutants have various origins, but since agricultural lands occupy 91% of the watershed, livestock operations, manure applications, fertilizer and mineralization of soil organic matter are believed to be the predominant sources.

The dominant soil association consists of Clarion (well-drained Typic Hapludolls), Nicollet (somewhat poorly drained Aquic Hapludolls), and Webster (poorly drained Typic Haplaquolls) soils (National Cooperative Soil Survey, 1986; Soil Survey Staff, 1999). The prairie potholes are occupied by very poorly drained Okoboji soils (Cumulic Haplaquolls), often with calcareous and poorly drained Harps soils (Typic Calciaquolls) on their margins. Most of the soils have a loam texture so in addition to sheet and rill erosion, slumping of streambanks during and after periods of high flow is another potential source of sediment pollution. Pastures occupy 6% of the watershed, most located along riparian valleys in the lower watershed where cattle then have free access to streams.

Corn (*Zea mays* L.) and soybean (*Glycine max* (L.) Merr.) are the predominant crops grown annually on 85% of the watershed area. This non-diverse cropping system (binary rotation) is common throughout the upper Mississippi River basin (Fig. 1) and is another factor that has affected soil and water quality. The 50-year decrease in land area devoted to hay (predominantly alfalfa (*Medicago sativa* L.)) and oat (*Avena sativa* L.) has reduced the portion of the year that soils are covered with living, transpiring plants, and significantly affected both the hydrology and soil resources (Schilling, 2005). The long-term change in cropping patterns throughout the South Fork watershed and entire Mississippi River basin coincides with changes in animal

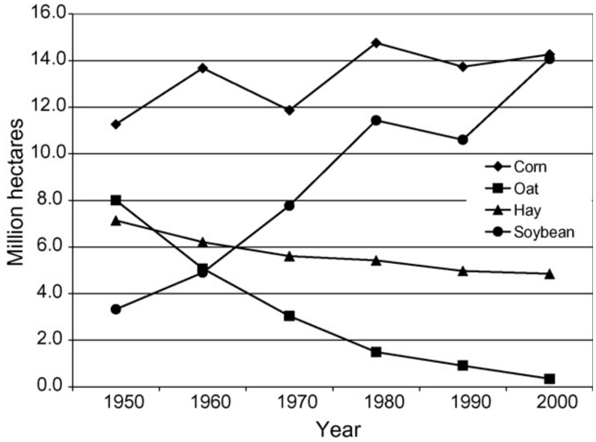

Fig. 1. Changes in crop production in the upper Mississippi River basin between 1950 and 2000 (National Agricultural Statistics Service (NASS), 2005).

production practices throughout the USA. For example, concentrated animal feeding operations (CAFOs) with most producing swine are now very common throughout the watershed.

1.2. Soil quality assessment strategies

Currently there is no specific protocol for evaluating soil quality at the watershed scale but two general approaches – surveys (e.g. Brejda et al., 2000; Cambardella et al., 1994) or paired comparisons (e.g. Moorman et al., 2004; Cambardella et al., 2004) have been suggested. Surveys will provide an overall assessment of soil quality within the watershed and be able to detect differences among sampling sites and/or landscape positions. Paired comparisons will enable the user to detect differences between specific soil management practices (e.g. till vs. no-till or manure vs. no-manure) and to develop correlations between soil quality response and specific treatment variables. We chose a survey approach to complement studies of land use and water quality that were already underway. Our intent was to determine how soil and crop management practices are affecting individual soil properties and overall soil quality within the South Fork watershed.

Both SMAF and the erosion model Revised Soil Loss Equation (RUSLE2) were used for our evaluations. The SMAF (Andrews et al., 2004) consists of three steps: indicator selection, indicator interpretation, and integration into a soil quality index. Currently, scoring curves have been developed for 11 potential indicators (i.e. soil properties), and the user chooses those most appropriate for the evaluation being made. For example, if the effects of adding manure are being evaluated, soil-test P would be an important indicator to include in the assessment. In the indicator interpretation step, measured or observed data are transformed into unitless scores based on the site-specific, algorithms developed for each soil function (e.g. productivity, environmental protection, or maximum waste disposal). For this assessment, the soil functions of interest include crop productivity, nutrient cycling, physical stability, water and solute flow, contaminant filtering and buffering and biodiversity. The integration steps allows for the individual indicator scores to be combined into a single index value. The tool's framework design allows researchers to continually update and refine the interpretations for different soils, climates, and/or land use practices.

RUSLE2 was developed jointly by the USDA-ARS, USDA-NRCS, and the University of Tennessee (Lightle, 2007; USDA-ARS, 2008). The computer program estimates soil loss from rill and interrill erosion caused by rainfall on cropland. It is used primarily to guide conservation planning, inventory erosion rates and estimate sediment delivery. The program has also been enhanced to compute soil tillage intensity rating (STIR), N-leaching, and the soil conditioning index (SCI) values.

The STIR rating was developed to replace the soil disturbance rating component of the SCI and to function as a stand-alone rating to evaluate tillage and/or planting effects on factors other than ground cover and surface residue distribution. STIR uses the operations database parameters in RUSLE2 to calculate a soil tillage intensity rating for the system used to grow a crop or for an entire rotation. The ratings are able to show differences between systems across the spectrum ranging from no-till to conventional moldboard plowing. Factors considered in the STIR ratings are (1) recommended operating speed, (2) tillage type, (3) tillage depth, and (4) surface area disturbed (USDA-NRCS, 2006). Higher STIR ratings are associated with greater soil disturbance and more frequent operations. Comparisons of STIR ratings for different tillage and planting systems provide insight into the carbon loss, moisture depletion, and air quality (dust) issues related to tillage of the soil.

The N-leaching index is computed based on soil hydrologic group, annual and winter precipitation (Pierce et al., 1991). It is a relative value that ranges from 0 to 25 and can be used to compare the potential for N-leaching among various management systems. Values of 0–2, 3–10, and >10 are considered to have low, medium, and high leaching potential, respectively (Lightle, USDA-NRCS, personal communication). Sandy soils that are more susceptible to leaching would be expected to have values approaching 25.

The SCI is a tool for organic matter prediction used by the Natural Resources Conservation Service (NRCS) (Hubbs et al., 2002). It helps quantify effects of climate, tillage, and erosion on organic matter decomposition and thus serves as an indicator of soil quality. Values are based on (1) the amount of organic material (OM) returned to the soil, (2) the effects of tillage and field operations (FO) on soil organic matter decomposition, and (3) the effect of predicted erosion (ER) associated with the management system. Qualitative changes in soil organic matter, with one of three outcomes – organic matter decline, organic matter increase, or organic matter equilibrium – are predicted using the equation:

$$\mathrm{SCI} = 0.4 \times \mathrm{OM} + 0.4 \times \mathrm{FO} + 0.2 \times \mathrm{ER}$$

where SCI is an index value that accounts for the combined effect of the three variables on SOM trends. Negative values indicate that the soil and crop management system that is being used is causing the soil organic matter levels to decrease and that practices should be changed to prevent further soil degradation.

The SCI calculation assumes the amount of biomass that must be returned to maintain equilibrium is directly proportional to the rate of decomposition. In moist climates, decomposition is more rapid than in dry climates, thus more biomass is needed. The same is true when comparing warm to cool climates. Maintenance amounts of crop residue at locations throughout the USA were calculated based on this assumption. Decomposition factors in RUSLE2 are used to estimate relative rates of plant residue decomposition at different locations.

While running RUSLE2 to compute SCI, users must input information about the field for which an assessment is to be made. This includes: (1) location (to determine climatic data), (2) soil texture, (3) site factors such as field slope length and steepness, (4) management sequence of all crops grown in the rotation, (including field operations such as tillage, fertilizer and manure application, harvesting, dates of operations, amount of irrigation, etc.), (5) applications of additional organic matter from sources such as manure or compost, and (6) information about supporting conservation practices such as strips/barriers, contouring, etc. Water erosion is computed in RUSLE2 and included in the SCI calculation. Wind erosion is not determined by RUSLE2. If wind erosion is an important factor where the evaluation is being made, it must be estimated by another method such as the Wind Erosion Equation (Woodruff and Siddoway, 1965). Wind erosion was not considered an important factor for our assessment which was designed to provide an initial assessment of soil quality within the South Fork watershed using various indicators and evaluation methods.

2. Materials and methods

Soil sampling for this study was conducted during autumn 2003 and spring 2004. Two transects, each 1.6 km in width, were established across the Iowa River South Fork watershed such that the major soil associations, landforms, crops, and sub-watersheds would be represented (Fig. 2). One 32-ha tract was randomly selected from each 259 ha (640 acre) section along each transect. Landowners and tenants were contacted for permission to collect soil samples and to obtain data on crop management history from each area.

2.1. Sample collection and analysis

Soil samples were collected by map unit from 29 of the 32 ha areas where permission was granted by the land owners and operators. Samples were not collected from areas without prior permission. Using NRCS soil maps, samples were collected by soil map unit (SMU) in each 32 ha area. Large areas of the same SMU were subdivided into approximately equal areas so that overall, each sample represented an area of approximately 3.6 ha (9 acres). This approach resulted in a total of 220 samples being collected for this study.

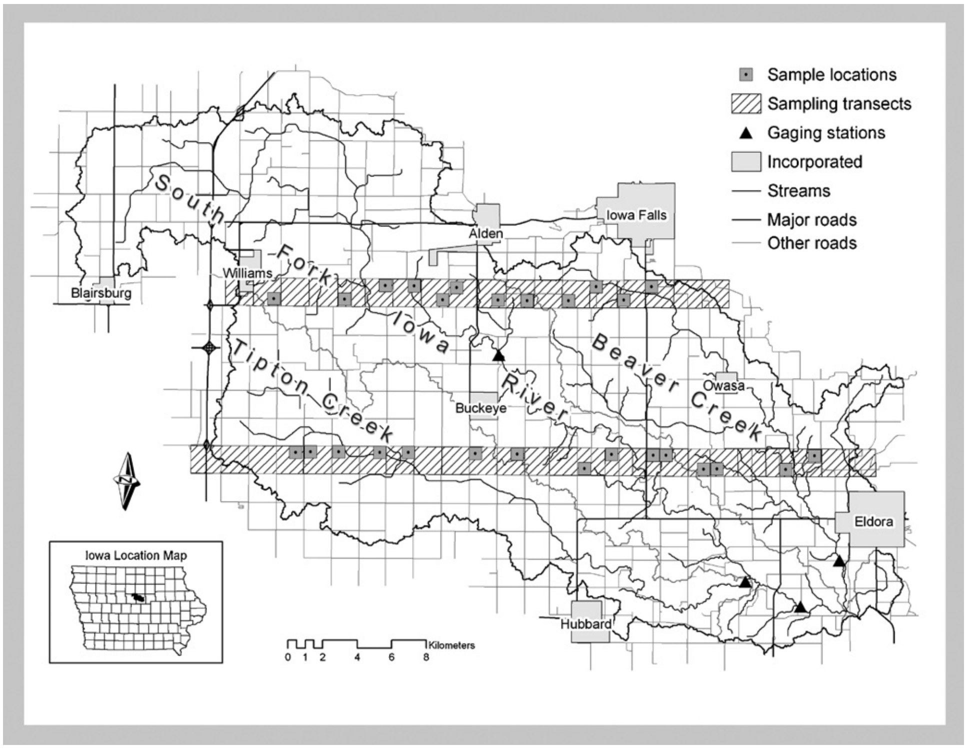

Fig. 2. Location of soil quality sampling transects within the South Fork watershed.

For each sampling site, 15–30 soil cores were taken to a depth of 15 cm using a soil probe with a 19 mm (3/4 in.) sampling tip. The samples were dried, ground and analyzed for pH using a 1:2 soil-to-water ratio (Watson and Brown, 1998), electrical conductivity (EC) (Whitney, 1998), Mehlich III extractable P, K, Ca, and Mg (Mehlich, 1984), total organic carbon (TOC) and total nitrogen (TN). Extractable P, K, Ca, and Mg concentrations were determined using an inductively coupled plasma-atomic emission spectrograph (ICP-AES). TOC and TN were determined by dry combustion with a Carlo-Erba NA1500 NCS elemental analyzer (Haake Buchler Instruments, Paterson, NJ).

Four of the indicators (pH, EC, TOC, and P) were used to compute a soil quality index (SQI) using the Soil Management Assessment Framework. The data for each sampling site were scored as outlined by Andrews et al. (2004) and used to compute indices for each site. The average slope for each 32 ha tract (not sampled map unit?) was determined from digital elevation maps so that the RUSLE2 could be run to estimate soil loss, N-leaching potential, STIR ratings, and the SCI for the soil series and management practices being used in each sampling area.

Mean, standard deviation, and simple correlations among the soil quality assessments (i.e. STIR, N-leaching, SCI, and SMAF ratings) for the entire dataset (220 sampling locations) were computed using SAS (SAS Institute, 2001) software. To determine if the assessments were strongly influenced by landscape position, tillage, crop rotation, or manure management, the dataset was sorted and reanalyzed using each of those factors as sampling groups. To analyze for landscape position, the SMUs were combined into four groups that described the landscape position (i.e. hilltop, sideslope, toeslope, or depression) where they were located, and were fit into these landscape groupings as follows. The Clarion, Clarion/Storden complex, and Zenor SMUs were classed as "hilltop" positions. The Nicollet, Saude, Lawler, Terril and Sparta SMUs were considered "sideslope" positions. The Webster, Webster-Nicollet complex, Coland, and Canisteo SMUs were considered "toeslope" positions, and the Okoboji, Harps, and Okoboji-Harps complex SMUs were grouped as depressions. This resulted in 86, 25, 68, and 41 observations, respectively, for each group. Note that the watershed's major soil association (Clarion-Nicollet-Webster), described above, includes several other soil types of lesser extent (National Cooperative Soil Survey, 1985) that were also included in the sampling, and comprised 49 of the 220 samples. The Canisteo series was the most common of these, comprising 28 of the 49 samples from these less extensive soils.

The data were examined using four crop sequences (i.e. continuous corn, corn–soybean, corn–corn–soybean, and meadow (which included the CRP sites)). The most common crop sequence was a 2-year corn–soybean rotation (71%) followed by a 3-year corn–corn–soybean rotation (15%) and continuous corn (11%). They were also analyzed according to the primary tillage practice (i.e. ridge-tillage, disking, chisel plow, and deep ripping), and according to manure management history histories (i.e. manure applied, no manure applied, or unknown).

3. Results and discussion

Our assessment showed that a 2-year corn–soybean rotation was the most common cropping practice (19 tracts) within the South Fork watershed at the time of sample collection. Three fields had a history of continuous corn and four had a 3-year corn–corn–soybean rotation (not shown in Fig. 2). One tract was planted to alfalfa and two were enrolled in the Conservation Reserve Program (CRP). With regard to assessment of soil quality, it is important to note that to have qualified for the CRP those areas would have to have a history of severe soil erosion.

From a nutrient management perspective, for which the survey was originally designed, TOC, pH, P, and K were the primary indicators. For the area surveyed within the watershed, the average values (Table 1) were very typical for a north central Iowa location. The average TOC was about one-half the value associated with native Midwestern mollisols which is typical of the change resulting during the 100–150 years since the prairie was first tilled. Soil pH was near neutral and extractable P and K concentrations were in the optimum range for corn and soybean production according to Iowa State University guidelines (Mallarino et al., 2002). Calcium and Mg concentrations were similar to those typically found in soils formed on the Des Moines Lobe in the northern Corn/Soybean Belt (Karlen et al., 2002).

Assessments using RUSLE2 showed that average soil loss was well below the “T” value for these soils, averaging 1.1 Mg ha^{-1} for our assessment sites. The leaching potential was low to medium, although a substantial portion of this watershed is tile drained (Tomer et al., in press-a) so careful N management is needed to prevent losses to streams and rivers draining the area (Dinnes et al., 2002). STIR ratings were also relatively low (averaging 68 $\pm$ 17) indicating conservation tillage was a predominant practice throughout the watershed. This was confirmed by Tomer et al. (in press-b) through a tillage-practices survey conducted in 2005.

Table 1
Overall assessment of soil quality indicators measured in 2003 and 2004 within the South Fork watershed of the Iowa River in north central Iowa, USA

Indicator	N	Mean	S.D.
Organic C (g kg^{-1})	220	28.4	13.6
Total N (g kg^{-1})	220	2.2	1.0
pH	221	7.0	0.8
EC (ds m^{-1})	220	0.32	0.14
P (mg kg^{-1})	221	36	29
K (mg kg^{-1})	221	162	53
Ca (mg kg^{-1})	218	5015	4330
Mg (mg kg^{-1})	218	450	330
Soil loss (Mg ha^{-1})	212	1.13	0.89
N-leaching index	212	3.3	1.8
STIR rating	212	68	17
SCI	213	0.4	0.2
SQI	220	87	8
TOC score	220	0.60	0.26
pH score	221	0.95	0.06
P score	221	0.96	0.11
EC score	220	0.96	0.11

Analysis using the SMAF showed that, overall, the soils were functioning for crop production at 87% of their estimated capacity. The lowest indicator score was for TOC (Table 1), because based on the scoring functions tailored for these soils in the SMAF (Andrews et al., 2004), the inherent levels had been depleted by approximately 40% since the prairies were first tilled.

3.1. Landscape effects

With regard to crop production, soil-test P ratings except in the depression areas were generally very high (>31 μg g^{-1}) (Mallarino et al., 2002), but not at levels anticipated to have a severe environmental impact (e.g. >100 μg g^{-1}). Lower soil-test P ratings in the depression areas were consistent with the higher pH in those soils (particularly the Harps series). Soil-test K was generally in the optimum range (131–170 μg g^{-1}), but the lower values are approaching levels that may result in early season K deficiencies if no-tillage practices are used (Karlen and Kovar, 2005) to reduce soil erosion. Both the SCI and SMAF ratings were also the lowest for the hilltop positions. The former is presumably driven by higher soil erosion rates and the latter by the low level of total organic carbon as a result of that erosion. RUSLE2-estimated soil loss was also the greatest on hilltops. Total organic C levels were lowest at hilltop positions and highest in the depression areas (Table 2), presumably reflecting water, wind, and tillage erosion which are well characterized for soils in this physiographic region (Schumacher et al., 2005). The individual indicator scores (Table 2) showed that TOC on hilltop positions had decreased by approximately 60% compared to the inherent values associated with the scoring functions in the SMAF. Soils in sideslope and toeslope positions were functioning at 60–66% with regard to TOC. The relatively high TOC score for the depression soils (Table 2) reflects the higher soil organic matter content in those soils because of the gradual accumulation of eroded sediments and naturally higher soil water content.

Total N values followed a similar pattern being lowest on hilltops and highest in the depression areas. Yet, C:N ratios at the hilltop and sideslope positions most frequently approached the 10:1 ratio (Fig. 3) where the risk of N saturation and nitrate leaching may be increased, a point well argued by Schipper et al. (2004). To be more specific, 46% of hilltop samples and 39% of sideslope samples had C:N ratios less than 12:1, whereas only 17% of toeslope and depressional soils had C:N ratios less than 12:1. Hilltop soils had a significantly greater C:N ratio than toeslope or depressional soils, based on Fisher’s LSD assuming unequal variances ($P < 0.10$), and sideslope C:N ratios were similarly distinguished from the depressions. Equivocally, RUSLE2-predicted N-leaching potential was highest for hilltops and sideslopes, averaging 4.9, compared to 1.5 for toeslope and depression areas (Table 2). The result that hilltop soils are most susceptible to both erosion and to leaching may seem counterintuitive, but a limited capacity to retain water and nutrients is an actual consequence of the long-term loss of topsoil (Lal et al., 2004). While we note that drainage tile with surface

Table 2
Soil quality indicators (mean and SD) in the South Fork watershed as affected by landscape group

Soil quality indicator	Landscape group			
	Hilltop (n = 86)	Sideslope (n = 25)	Toeslope (n = 68)	Depression (n = 41)
Organic C (g kg^{-1})	18.9 (4.9)	24.1 (5.4)	30.8 (6.9)	47.1(17.2)
Total N (g kg^{-1})	1.53 (0.38)	1.96 (0.40)	2.39 (0.53)	3.35 (1.56)
pH	6.6 (0.6)	6.4 (0.7)	7.1 (0.7)	7.8 (0.5)
EC (ds m^{-1})	0.24 (0.10)	0.26 (0.12)	0.36 (0.12)	0.44 (0.12)
P (mg kg^{-1})	38 (30)	45 (32)	38 (30)	22 (15)
K (mg kg^{-1})	154 (52)	165 (50)	164 (50)	172 (63)
Ca (mg kg^{-1})	3126 (2665)	3073 (1206)	5209 (2874)	9965 (6196)
Mg (mg kg^{-1})	380 (308)	379 (105)	531 (452)	508 (138)
Soil loss (Mg ha^{-1})	1.7 (1.0)	1.1 (0.8)	0.8 (0.4)	0.4 (0.2)
N-leaching index	5.0 (0.1)	4.9 (1.1)	1.7 (1.0)	1.4 (0. 1)
STIR rating	68 (15)	69 (11)	69 (2)	66 (21)
SCI	0.3 (0.2)	0.4 (0.2)	0.5 (0 1)	0.4 (0.2)
SQI	82 (7)	87 (8)	89 (5)	94 (6)
TOC score	0.40 (0.18)	0.60 (0.21)	0.66 (0.19)	0.93 (0.14)
pH score	0.98 (0.02)	0.98 (0.02)	0.95 (0.06)	0.89 (0.08)
P score	0.95 (0.13)	0.98 (0.05)	0.97 (0.07)	0.92 (0.13)
EC score	0.94 (0.15)	0.91 (0.17)	0.99 (0.04)	0.99 (0.03)

inlets are often placed in the depression areas, thus short-circuiting the natural drainage processes (Dinnes et al., 2002, Tomer et al., in press-a), soil carbon largely determines the capacity of soil to retain nutrients against leaching, and this capacity is clearly most limited in the upper landscape classes.

3.2. Tillage effects

The dominant primary tillage practice throughout the survey area within the South Fork watershed was chisel plowing (50%) with disking (24%) and deep ripping (11%) being the next most common practices. Ridge-tillage (5%) was represented, but this practice was used by only one operator in the entire watershed (Tomer et al., in press-b). Mean soil quality indicator values for the various tillage practices are presented in Table 3. Total organic carbon and pH were highest in areas where deep tillage was used, possibly because deep tillage is one response to compaction problems that are most common in low-lying areas

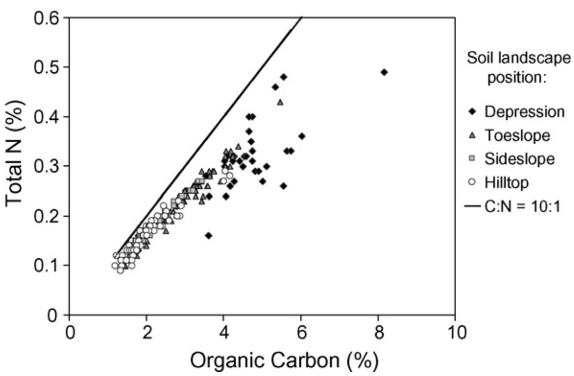

Fig. 3. Total organic C plotted against total N contents for soils sampled from four landscape positions in the South Fork of the Iowa River watershed. Soils most closely approaching the 10:1 line are dominantly hilltop positions, and, according to Schipper et al. (2004) are at greatest risk of N saturation leading to nitrate leaching.

with wetter soil conditions, and/or because farmers avoid deep tillage in higher landscape positions more susceptible to erosion. The TOC score was also the greatest for the deep tillage treatment. The SCI values for the sites sampled in this survey generally averaged 0.4 even though the SMAF assessment indicated TOC was the most impaired of the four indicators measured. Since the SMAF assessment uses measured data this indicates more calibration may be needed to further refine SCI for northern Corn/Soybean Belt soils.

3.3. Cropping system effects

Among the 29 fields surveyed, only 3% (seven sampling sites) were in either CRP or long-term meadow. Of the seven sites, two were located on highly eroded areas (Clarion/Storden Complex with 5–9% slope) and two were in areas with a seasonally high water table. The soil quality indicators (Table 4) showed relatively low TOC, a lower TOC score, and the lowest soil-test P levels among the crop sequence groups. We suggest this occurred because two of the three meadow areas were enrolled in the CRP, suggesting that the low TOC and P values reflect the fact that such areas were often highly eroded before they were enrolled and that they had lost a substantial portion of topsoil, TOC, and phosphorus fertility. Soils under CRP plantings are often in a state of soil reclamation/reconstruction. As expected STIR ratings and current estimated soil erosion were lowest in the areas planted to meadow due to the permanent cover. The lower SQI rating for the meadow group is attributed to low soil-test P and C levels (Table 4), which is also reflected by slightly lower TOC and P scores for those indicators.

3.4. Animal manure effects

Three manure management histories (i.e. manure applied (47%), no manure applied (28%), and unknown (25%)) were identified during the survey process. Among the soil property measurements, differences were gen-

Table 3
Soil quality indicators (mean and S.D.) in the South Fork watershed as affected by tillage practice

Soil quality indicator	Tillage practice			
	Ridge-till ($n = 10$)	Chisel plow ($n = 111$)	Disk ($n = 51$)	Deep-till ($n = 25$)
Organic C (g kg^{-1})	26.5 (8.9)	29.2 (15.7)	26.6 (10.0)	32.8 (11.0)
Total N (g kg^{-1})	2.14 (0.57)	2.25 (1.24)	2.05 (0.74)	2.47 (0.74)
pH	7.0 (0.7)	6.9 (0.8)	6.8 (0.7)	7.3 (0. 8)
EC (ds m^{-1})	0.33 (0.17)	0.30 (0.13)	0.36 (0.14)	0.39 (0.14)
P (mg kg^{-1})	30 (14)	36 (30)	43 (32)	42 (31)
K (mg kg^{-1})	124 (22)	164 (57)	166 (51)	184 (50)
Ca (mg kg^{-1})	4249 (2578)	4990 (4515)	4409 (3172)	5633 (3676)
Mg (mg kg^{-1})	432 (127)	433 (225)	425 (144)	408 (88)
Soil loss (Mg ha^{-1})	1.3 (0.3)	1.1 (0.8)	1.2 (1.0)	0.9 (1.0)
N-leaching index	3.2 (1.9)	3.4 (1.8)	3.3 (1.8)	2.6 (1.7)
STIR rating	38 (0)	70 (11)	76 (11)	69 (11)
SCI	0.4 (0.1)	0.4 (0.1)	0.4 (0.1)	0.40 (0.2)
SQI	88 (6)	87 (7)	87 (7)	92 (4)
TOC score	0.59 (0.24)	0.61 (0.26)	0.58 (0.25)	0.74 (0.22)
pH score	0.97 (0.03)	0.96 (0.06)	0.96 (0.06)	0.95 (0.04)
P score	0.96 (0.06)	0.95 (0.13)	0.96 (0.08)	0.95 (0.10)
EC score	0.99 (0.04)	0.97 (0.08)	0.92 (0.19)	0.98 (0.07)

Table 4
Soil quality indicators (mean and S.D.) in the South Fork watershed as affected by crop sequence

Soil quality indicator	Cropping sequence group			
	Corn–corn ($n = 24$)	Corn–soybean ($n = 157$)	Corn–corn–soybean ($n = 33$)	Meadow ($n = 7$)
Organic C (g kg^{-1})	29.0 (10.4)	27.0 (11.7)	35.1 (21.3)	28.2 (11.2)
Total N (g kg^{-1})	2.23 (0.70)	2.08 (0.77)	2.72 (1.88)	1.93 (0.64)
pH	7.0 (0.9)	6.9 (0.7)	7.1 (0.8)	7.2 (0. 7)
EC (ds m^{-1})	0.38 (0.14)	0.30 (0.14)	0.37 (0.14)	0.30 (0.07)
P (mg kg^{-1})	48 (34)	36 (29)	31 (27)	15 (14)
K (mg kg^{-1})	196 (48)	161 (53)	139 (47)	157 (50)
Ca (mg kg^{-1})	5021 (2828)	4893 (4511)	5489 (4666)	5526 (3186)
Mg (mg kg^{-1})	438 (147)	450 (379)	449 (147)	488 (258)
Soil loss (Mg ha^{-1})	1.1 (1.0)	1.2 (0.9)	1.0 (0.4)	0.5 (0.8)
N-leaching index	2.7 (1.8)	3.4 (1.8)	3.1 (1.8)	3.4 (1.9)
STIR rating	85 (0)	69 (10)	68 (23)	10 (23)
SCI	0.3 (0.1)	0.4 (0.1)	0.4 (0.1)	0.8 (0.3)
SQI	89 (7)	86 (8)	89 (7)	82 (14)
TOC score	0.64 (0.26)	0.58 (0.27)	0.71 (0.24)	0.60 (0.29)
pH score	0.96 (0.03)	0.96 (0.06)	0.93 (0.10)	0.94 (0.08)
P score	0.98 (0.05)	0.95 (0.12)	0.96 (0.10)	0.89 (0.18)
EC score	1.0 (0)	0.95 (0.14)	0.98 (0.06)	1.0 (0)

Table 5
Soil quality indicators (mean and S.D.) in the South Fork watershed as affected by manure history

Soil quality indicator	Manure history		
	Applied routinely ($n = 103$)	Never applied ($n = 61$)	Unknown ($n = 56$)
Organic C (g kg^{-1})	30.5 (15.2)	25.1 (10.3)	28.4 (13.2)
Total N (g kg^{-1})	2.34 (1.22)	1.90 (0.71)	2.22 (0.84)
pH	7.1 (0.8)	6.7 (0.7)	7.0 (0.8)
EC (ds m^{-1})	0.34 (0.14)	0.33 (0.12)	0.28 (0.15)
P (mg kg^{-1})	42 (35)	32 (22)	28 (19)
K (mg kg^{-1})	169 (53)	165 (56)	145 (50)
Ca (mg kg^{-1})	5096 (3907)	4402 (3560)	5538 (5623)
Mg (mg kg^{-1})	425 (140)	395 (137)	554 (599)
Soil loss (Mg ha^{-1})	1.1 (0.9)	1.2 (0.9)	1.0 (0.9)
N-leaching index	3.1 (1.8)	3.6 (1.8)	3.3 (2.0)
STIR rating	79 (16)	57 (17)	62 (0.4)
SCI	0.4 (0.1)	0.5 (0.2)	0.4 (0.2)
SQI	88 (7)	86 (8)	86 (9)
TOC score	0.64 (0.26)	0.53 (0.26)	0.61 (0.27)
pH score	0.96 (0.11)	0.97 (0.03)	0.95 (0.06)
P score	0.98 (0.07)	0.94 (0.12)	0.94 (0.15)
EC score	0.97 (0.12)	0.97 (0.09)	0.93 (0.14)

Table 6
Correlation coefficients among soil quality indicators used within the South Fork watershed

Variable	SCI	Soil loss	N-leaching index	STIR	SQI
SCI	1.00				
Soil loss	−0.70[a]	1.00			
N-leaching index	−0.51[a]	0.53[a]	1.00		
STIR	−0.53[a]	0.10	−0.03	1.00	
SQI	0.29[a]	−0.46[a]	−0.51[a]	0.08	1.00

[a] Indicates significance at $P < 0.0001$.

erally small among the three groups, although areas with a history of manure application did have soil-test P values that are considered "very-high" for crop production purposes (Table 5). The P levels, however were not high enough to be penalized as a potential environmental hazard by the SMAF analysis.

3.5. Correlations among soil quality indicators

In addition to computing mean values for the various soil quality indices (i.e. Soil loss, STIR, SCI, N-leaching, and SQI), we also determined correlation coefficients between the various indices (Table 6). The SCI and soil loss, N-leaching index, and STIR ratings were negatively correlated at $P < 0.0001$. The SQI was also negatively correlated at $P < 0.0001$ with soil loss and the N-leaching index but showed no significant relationship to the STIR rating. The latter was somewhat surprising because for many soil quality indicators a negative relationship (Andrews et al., 2004) would be expected because more intensive tillage increases oxidation of soil organic matter, fractures aggregates into smaller pieces, depletes soil water, and increases the potential for fugitive dust (i.e. lower air quality). Soil loss and N-leaching showed a highly significant ($P < 0.0001$) correlation, presumably because RUSLE2 would calculate greater soil loss and leaching potential from hilltops than other landscape positions. The correlation between SCI and SQI was also highly significant ($P < 0.0001$). This is important because even though improvement can undoubtedly be made for both assessment tools, it does indicate both tools are providing consistent assessment information about various soil and crop management practices for this watershed.

4. Summary and conclusions

This study provides a preliminary assessment of how various tillage and crop management practices are affecting soil quality indicators at various landscape positions within the Iowa River's South Fork watershed. Both the SCI derived using the RUSLE2 erosion model and a SQI computed using the SMAF indicate that soil quality in this watershed is relatively stable for the management practices being used. Areas with a history of manure application (primarily swine manure) have soil-test values that are considered "very-high" for crop production purposes, but for the fields we had permission to sample, they are not yet at a level that would be detrimental to water quality. However, water quality monitoring in the watershed shows stream P concentrations often exceed eutrophication risk thresholds (Tomer et al., in press-a), therefore our results may not represent P contents of all manured soils across the watershed. Soil organic C levels and the C:N ratios at the upper landscape positions appear to be approaching critical levels where the risk of N saturation and nitrate leaching may be increasing. N-leaching is a significant concern in this watershed (Tomer et al., in press-a) and results of this assessment indicate that soil management in upper landscape positions may be as important to addressing this issue as the lower lying, tile drained soils.

The SCI and SQI both showed the importance of maintaining or increasing soil organic C with regard to soil quality in the South Fork watershed of the Iowa River. This may suggest that increased use of reduced or no-tillage practices would be beneficial in order to increase soil C levels. Tomer et al. (in press-b) reported that 29% of this watershed's agricultural land was conventionally tilled, while only 7% was managed in no-tillage, which suggests opportunities for improved soil management certainly do exist. During transition to decreased use of tillage, close monitoring of soil-test K is recommended to prevent that essential plant nutrient from limiting crop yields.

The SCI and SMAF are both useful for assessing effects of various soil management practices. The SMAF uses measured data and can return site-specific assessments for more factors (e.g. soil-test P, pH, and TOC), but collecting and analyzing that data does increase the overall cost. We conclude that our assessment approach was successful in pointing out several soil management issues in the watershed, which combined with water quality monitoring results helps link the effects of agricultural management practices with impacts on both soil and water resources. Undoubtedly, further assessments at this scale are needed, but the basic approach seems appropriate and consistent with the goals stated in the Soil and Water Quality: An Agenda for Agriculture publication (NRC, 1993).

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